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Chino Mines Company
Administrative Order on Consent
Lampbright Investigation Unit
Human Health Risk Assessment Work Plan

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For:

New Mexico Environment Department

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Executive Summary

A baseline human health risk assessment (HHRA) will be conducted by Neptune and Company, Inc. (Neptune) for the Chino Mines Company (Chino) Lampbright Investigation Unit (LIU; near Hurley, New Mexico) to evaluate the potential for adverse human health effects associated with historical mining-related contamination. This Work Plan describes the proposed process to be followed, and satisfies Task 1 of the Consulting and Professional Services agreement between Neptune and Chino. Task 2 is described under “Data Evaluation for Constituents of Interest” below (note that Task 1 and 2 were funded for 2011). Task 3 will involve the HHRA activities described in this Work Plan.

The HHRA will focus upon providing the best information possible to make informed and expedient decisions regarding the LIU. The HHRA will assist the involved parties (New Mexico Environment Department [NMED], Chino, and the public) in making decisions regarding remediation and risk management at the site, in accordance with the Administrative Order of Consent (AOC) entered on December 23, 1994. A site-wide ecological risk assessment is being conducted as a separate effort by another contractor.

The Remedial Investigation (RI) report for the LIU (Arcadis, 2011) contains a substantial amount of historical and background information. This information will not be repeated here except as necessary for context. The LIU HHRA will be limited to metal compounds, per prior agreement between NMED and Chino (Chino, 2010), and will be largely based upon environmental data collected during the RI and previous investigations.

The proposed HHRA described in this Work Plan will incorporate a two-tiered approach. The screening-level Tier I assessment will assess maximum detected concentrations of chemical constituents in exposure equations that include conservative (i.e., biased toward protection of human health) exposure and chemical toxicity assumptions. This assessment will identify constituents of interest (COIs) to be carried forward to the Tier II assessment. A screening assessment was performed in the RI using EPA soil screening criteria, but these criteria do not encompass all potential exposure pathways for the LIU. Therefore, the Tier I assessment will include all potentially applicable exposure pathways. The Tier II assessment will involve statistical estimation of exposure point concentrations (EPCs) of COIs and refined exposure assessment models. If necessary and informative, the Tier II assessment may be expanded to include probabilistic analysis to identify important sources of variable uncertainty.

The HHRA will inform decisions as to whether unacceptable human risks associated with the LIU under current and future land uses may exist, as well as risk management or remediation decisions. Remedial Action Criteria (RAC) development is not specifically addressed in this Work Plan, as such development will only occur in a dispute resolution scenario, and will be addressed via separate tasks negotiated between Neptune, NMED, and Chino.

1 Introduction

1.1 Context

Environmental investigation at the overall Chino site is governed by the complex regulatory milieu. The LIU is one of six IUs within the AOC Investigation Area (Arcadis, 2011; Figure 1-1), which are all subject to distinct HHRA. The RI process in general is conducted under the AOC between Chino and NMED. The AOC (effective December 23, 1994) addresses effects of historical operations from Chino's copper mining and processing facilities within the AOC Investigation Area. Therefore, the AOC is the most relevant and direct regulatory structure for the HHRA.

In addition to the AOC, New Mexico Water Quality Control Commission regulations require development of a discharge plan for discharge of effluent or leachate to prevent adverse impact on groundwater resources (Chino, 1995). Many of the activities to be addressed under the AOC for LIU are being addressed under discharge permit (DP)-related programs (i.e., Sitewide Abatement and the DP-376 Corrective Action). Based upon prior agreement between Chino and NMED, the LIU RI Report will focus on issues outside of those covered under DP-related programs.

In addition to the discharge plan, Chino is also regulated under Clean Water Act regulations for stormwater discharges via EPA's Multi-Sector General Permit program. However, Chino collects and manages all stormwater that contacts stockpiles and tailing impoundments on site. Thus, these facilities are effectively zero-discharge for surface water. Only certain areas at Chino are authorized to discharge storm water, and these areas are limited to plant operation and maintenance areas, access roads and material storage areas located outside of Chino's zero-discharge area.

1.2 Purpose and Objectives

This Work Plan describes the proposed technical approach that will be employed in the LIU HHRA. The basic approach is that defined by the United States Environmental Protection Agency's (EPA's) Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A (EPA, 1989). The components of this approach involve data collection and evaluation, exposure assessment, toxicity assessment, and risk characterization.

The proposed HHRA described in this Work Plan will incorporate a two-tiered approach. A Tier I assessment involves screening of COIs using maximum detected constituent concentrations and highly conservative assumptions. A type of screening assessment was performed in the RI, but will be expanded for the purpose of the HHRA. The Tier II assessment will focus upon the specific COIs that are the primary sources of potentially unacceptable human health risks (based upon the Tier I assessment), and will involve refined receptor (i.e., a type of person exposed at the site) and exposure-pathway specific calculations. These assessments will inform decisions as to whether unacceptable human risks associated with the LIU under current and future land uses may exist, as well as any risk management or remediation decisions.

The HHRA will focus upon providing the best information possible to make informed and expedient decisions regarding the LIU. The Tier I and initial Tier II components of the HHRA will be ‘deterministic’; i.e., they will estimate single-point values for risk in different scenarios, as opposed to a ‘probabilistic’ analysis, which estimates distributions or ranges of values via incorporation of variability and uncertainty associated with assumptions. If potentially unacceptable risks are found in the initial Tier II HHRA, then it may be informative to conduct a more detailed probabilistic (i.e., Monte Carlo simulation) assessment to identify the degree of conservatism associated with the Tier II assessment and to identify important sources of uncertainty. This may also include further analysis of site data.

The following sections describe the essential components of a HHRA:

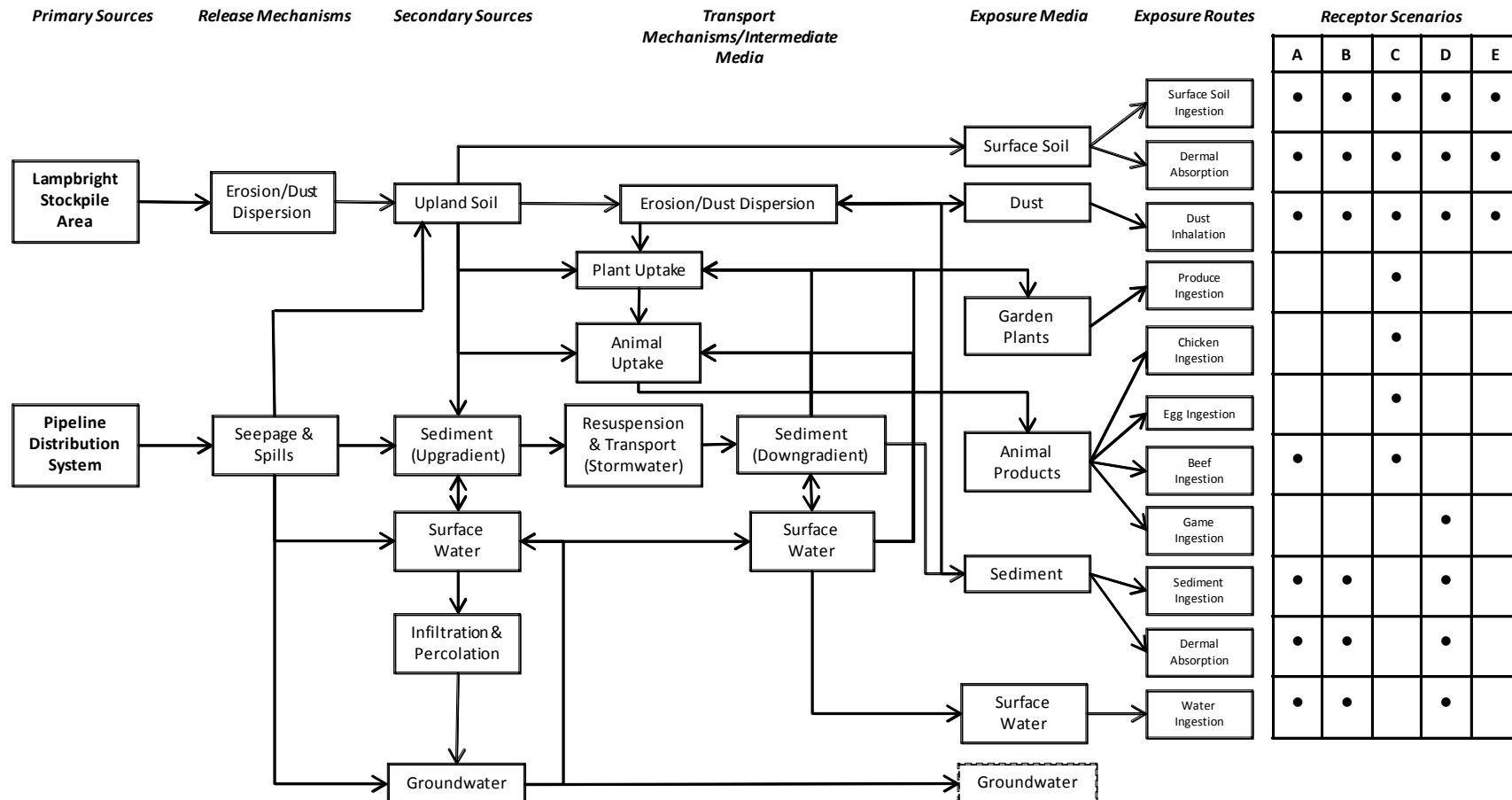
- Conceptual site model (CSM);
- Data evaluation for COIs;
- Exposure assessment;
- Toxicity assessment;
- Risk characterization;
- Uncertainty assessment; and,
- Recommendations and conclusions.

2 Conceptual Site Model

2.1 Overview

The CSM functions as a tool for integrating COI sources, release mechanisms, secondary sources, transport mechanisms, intermediate exposure media, final exposure media, exposure routes, and receptors. The CSM forms the overall framework for the HHRA. The major components of the CSM are a physical model of the IU and an exposure model. The RI (Arcadis, 2011) contains figures indicating the location and scale of the site (e.g., Figure 1-1). Figure 1 below is a graphical representation of the proposed CSM.

Figure 1: CSM for the LIU HHRA



Notes:

Receptor Scenarios defined in Table 1.

Groundwater (dashed boxes) is not addressed in the HHRA. See text for explanation.

2.2 Physical System Model

2.2.1 Description

The purpose of developing a model of the physical system is to define the key processes and features in the environment that are believed to control COI distribution within the LIU. This includes all the processes in the CSM leading up to exposure concentrations in final exposure media (i.e., COI sources, release mechanisms, secondary sources, transport mechanisms, and intermediate exposure media).

2.2.2 Types of Contamination

Contamination to be addressed in the HHRA includes metals associated with historical mining operations and releases. Although other types of chemicals were and are employed in mining operations (e.g., raffinate), these chemicals are typically captured to the extent possible for re-use, and any water releases are covered under the DP. Thus, NMED and Chino have agreed to focus upon metals for the RI and the HHRA.

2.2.3 Lampbright Investigation Unit Description

The LIU is located in the northeast corner of the overall AOC Investigation Area, south of State Highway 152 and east of the operational Santa Rita Open Pit (see Figures 1-1 and 2-1 in Arcadis, 2011). The LIU includes the area surrounding the present Lampbright Leach Stockpile that may be affected by historic operations. Specifically, it includes “Tributary 1” downgradient of Dam 8 (which forms Reservoir 8, a pregnant leach solution [PLS] collection area), the North Cut Diversion Area, and “Tributary 2” plus other downgradient areas. Note that these drainages are generally ephemeral, with flow occurring only during storm and spring runoff events (however, pools may exist for extended period). Surrounding upland areas are also included in the LIU.

2.2.4 Contamination Sources, Releases, and Transport

There are a variety of potential sources and release mechanisms of COI contamination associated with the LIU that were investigated in the RI. These are roughly categorized as primary and secondary.

Primary sources of environmental releases include the Lampbright Stockpile Area (LSA), the solution extraction/electrowinning (SX/EW) plant, and the PLS system, including pipelines, collection tanks, and reservoirs. Primary release mechanisms include fugitive dust, spray (from raffinate emitters on the LSA), rainwater seepage, spills, and storm water events. Transport mechanisms include infiltration and percolation in the LSA, overland flow of contaminated water, resuspension via stormwater, and accidental spills of PLS and process water. Secondary sources include upland soil impacted by fugitive dust, tributary sediment impacted by dust or runoff, and biotic (i.e., plant and animal) uptake.

Potential groundwater impacts are not to be evaluated in the HHRA, as these are addressed under a different regulatory construct; i.e., Discharge Plan (DP) 376/Corrective Action and Site Wide Abatement. The RI report (Arcadis, 2011) contains details. Any exceedences of groundwater criteria would be addressed under that regulatory construct. The groundwater data are relevant, however, for the LIU HHRA as sediment data have

been collected under the DP. Transport mechanisms include potential interfaces of groundwater and surface water plus sediments.

Factors affecting the extent of present contamination include changes in the footprint of the LSA, specific extraction processes, and the nature and design of collection systems. Additionally, there have been past remediation efforts at the LIU. Three major historical PLS releases have occurred; in 1985, 1988, and 2007. The first two releases affected Tributary 1, and the most recent Tributary 2. Remediation and monitoring related to these spills were addressed under DP-376. A major remediation effort was conducted for the 2007 event in which a large amount of sediment and surface water was removed. Contamination from this event reportedly did not extend beyond the confluence of Tributary 1 and 2. The RI report (Arcadis, 2011; Section 2.8.8) contains further details.

2.2.5 Exposure Point Concentrations

The HHRA will rely upon data collected in the RI, as well as any other relevant data, to estimate EPCs for COIs that are identified by the Tier I screening assessment. EPCs are single values, typically means (averages) and 95% upper confidence limits (UCLs) on means, which are employed along with exposure assumptions and equations (described below) to estimate receptor exposures. As the COIs at this site are naturally-occurring and the Chino mine area is heavily mineralized, comparisons with relatively non-impacted 'background' or 'reference' areas will be made to provide perspective (see further discussion below). In some cases further modeling is required to estimate EPCs. For example, exposure to fugitive dust associated with contaminated soil requires use of a simple model to convert soil concentrations into respirable (i.e., able to be inhaled deep into the lungs) dust concentrations.

2.3 Exposure Models

2.3.1 Description

Exposure models are qualitative and quantitative (i.e., equations) means to 'translate' EPCs into estimates of receptor exposure. These estimates are in turn combined with estimates of COI toxicity to estimate risks.

In general, exposure models incorporate assumptions regarding:

- Types of receptors;
- Characteristics and behavior of those receptors; and,
- Likely areas where receptors will live, work, recreate, and so forth; and how these intersect with the spatial extent of contamination.

These assumptions are collectively termed 'scenarios'. Typically, such scenarios are based upon current and likely future land use. At an operating site such as this, future land use is difficult to predict, as this is highly dependent upon the market for the mine's product, other economic factors, population pressures, changing demographics, and so forth. For example, demand for the mine's product may go down in the future, and thus the work force may be reduced; but the area may become more desirable for retirees. Prediction of land use post-closure is even more difficult. Therefore, the LIU HHRA will

only evaluate the most likely scenarios, based upon current use and observation of land use in surrounding areas.

In the type of HHRA proposed here, exposure is typically estimated for a hypothetical receptor under ‘reasonable maximum exposure’ (RME) conditions. The intent of the RME concept is to ensure that it is likely that exposures and risks will be overestimated, as opposed to underestimated; but not to evaluate absolute worst-case conditions. There are two ways that RME is typically defined:

- A. By defining exposure scenarios that include assumptions and activities that would result in a comparatively large degree of exposure. Typically this involves allowing receptors to live on the site (in the case of the LIU, this would involve a typical rural residential lifestyle), as well as other activities; and,
- B. By including a number of conservative assumptions (e.g., 95th percentiles of population distributions) in exposure models (e.g., a receptor who breathes at a high rate, drinks a lot of water, etc.) with 95% UCL EPCs.

Both of these RME ‘methods’ will be employed in the HHRA. In the case of B above, qualitative judgments are made regarding upper-bound estimates to result in RME, as opposed to worst-case, estimates. Additionally, deterministic HHRA (e.g., previous HHRA at the Chino site) often employ ‘central tendency’, ‘average’, or ‘typical’ assumptions as a point of comparison. This will be addressed in the LIU as germane to A above by including current use (e.g., ranching). In the case of B above, the difficulty lies in determining the proper combination of values that actually result in an ‘average’ exposure. Additionally, the degree of conservatism associated with deterministic RME exposure model results may be unknown, so comparisons between an ‘average’ estimate and a RME estimate do not provide an accurate estimate of the degree of bias associated with the RME estimate. Basically, there is a large degree of confidence that a RME estimate is conservative, but the degree of conservatism is unknown.

Therefore, for the purpose of this HHRA, an alternative method is proposed. RME exposures will be estimated in the usual fashion. If unacceptable risks are found in the Tier II assessment, and/or if the involved parties are not comfortable with the degree of uncertainty associated with risk estimates, then a simple probabilistic analysis (Cullen and Frey, 1999) will be conducted only for those COIs and exposure pathways that are problematic. This will allow a much more detailed and accurate representation of uncertainties than is provided by a comparison between RME and ‘average’ exposure/risk estimates. Additionally, the probabilistic analysis will allow determination of the degree of conservatism associated with RME estimates; e.g., whether a RME estimate represent a 90th percentile, a 99th percentile, and so on. This will provide the involved parties the best information for decision-making.

2.3.2 Exposure Scenarios

The most likely generic scenarios, based upon current land use, previous HHRA (i.e., for the Hanover Whitewater Creek and Smelter Tailings Soils IUs; see Neptune, 2008, and Gradient, 2008), a tour of the overall site, and discussions with Chino and NMED, include:

-
- Present and future ranching;
 - Present trespassing on Chino property;
 - Future residential development;
 - Future recreation (e.g., hiking, hunting, off-highway vehicle [OHV] riding); and,
 - Future construction work.

The land in the vicinity of Tributaries 1 and 2 is presently owned by Chino Mines and leased for cattle grazing. Access to this area for the general public via Highway 152 and other roads is feasible, but limited. Based upon interviews with Chino staff, there appears to be little current recreational use, probably because the area is largely fenced and use beyond the approved ranching would be considered trespassing. There are no current residences on the property, although the nearest is only 1 km from the eastern LIU boundary. Evaluation of potential future land use will follow precedent set in the HWC and STS IUs HHRA's (Neptune, 2008; Gradient, 2008). In both IUs, areas that are presently owned by Chino but could feasibly support future development were evaluated under a range of land use options.

Some exposure-related activities may be predominantly associated with specific geomorphic (i.e., land and geology) features and locations. For example, cattle grazing may be associated primarily with areas that have more fertile soil and thus support plant growth. Fugitive dust exposure may be associated primarily with unvegetated geomorphic features having a higher proportion of fine particulates. Residences may be limited by terrain, proximity to roads, and other considerations. To the extent possible, these factors will be considered in the HHRA. In the simplest sense, a distinction will be made between exposures over the entire IU (including both uplands and tributaries) versus exposures to only upland areas.

Receptors may be exposed to COIs via ingestion, inhalation, and dermal (skin) absorption. Ingestion may include dust/soil, plants that have taken up COIs via the soil, animals that ingest dust/soil and contaminated plants, and water. Inhalation involves breathing in dust, and dermal absorption involves COIs being deposited and absorbed into the skin.

While it is possible that future residents could drill for and drink groundwater at the LIU, groundwater is addressed under the regulatory structure of the DP. COIs are monitored in a number of wells, and any exceedences of groundwater criteria trigger regulatory action under the DP. As this situation is likely to continue for the foreseeable future, groundwater will not be explicitly addressed in this HHRA.

It is possible that some receptors might drink surface water at the LIU. Surface water is generally ephemeral in the tributaries, but pools can persist. Ranchers and hunters, for example, could occasionally drink this water; assuming treatment for microorganisms. The RI (Arcadis, 2011) found that there were no exceedences of drinking water criteria at the site; however, this does not exclude the possibility that risks may be present. Surface water may also be an important transport medium.

In the HWC and STS IU HHRA (Neptune, 2008; Gradient, 2008), food pathways (e.g., home-grown produce, chickens, beef, etc.) were evaluated, and found to be important in terms of contribution to total estimated exposures. For this reason, the assumptions employed in these calculations will be examined to estimate accurate exposures from these pathways in the LIU HHRA. Additionally, plant data collected for the purpose of ecological risk assessment (Newfields, 2005) will be examined as to relevance and applicability.

Table 1 describes specific exposure scenarios to be evaluated in the LIU HHRA:

Table 1: Exposure Scenarios

Scenario	Receptor Type	Age	Examples of Activities	Location of Activities	Exposure routes	Notes
A: Ranching	Ranch-hand	Adult	Herding cattle, mending fences, riding OHV	Upland and tributaries	Dust inhalation, soil ingestion, beef ingestion, dermal absorption, surface water ingestion	<i>Present and future</i> Assumed to live off the LIU site
B: Trespassing	Local resident	Adult	Walking, shooting	Upland and tributaries	Dust inhalation, soil ingestion, dermal absorption, surface water ingestion	<i>Present</i> Assumes that young children would not trespass
C: Residence (acreage)	Family	Adult, child (0-6 years)	Living in house, playing in yard, walking on property, riding OHV	Upland	Dust inhalation, soil ingestion, produce ingestion, home-grown meat ingestion, dermal absorption	<i>Future</i> Assumes low probability that a house would be built in a tributary due to flooding risk
D: Recreation	Local resident	Adult	Riding OHV, hunting	Upland and tributaries	Dust inhalation, soil ingestion, game ingestion, dermal absorption, surface water ingestion	<i>Future</i> Assumes that a hunter riding an OHV would be a highly-exposed recreationalist
E: Construction	Local resident	Adult	Digging, operating machinery, construction work	Upland	Dust inhalation, soil ingestion, dermal absorption	<i>Future</i> Assumes low probability that a house (or other building) would be built in a tributary due to flooding risk

These categories of receptors will be defined in more detail in the HHRA. Note that children are only evaluated in the residential scenario, but this may change depending upon the level of detail necessary. “Children” are defined as from birth to 6 years, and specifically ages 3 to 6 in terms of soil ingestion (EPA, 2011), as this age group tends to engage in hand-to-mouth activities that result in more soil ingestion. This essentially

defines this age group as a special population, as soil ingestion often is a driving factor in soil-related risk. Otherwise, “adult” exposure variable values are generally defined here as those relevant to ages 16 and older. Ages in-between are not evaluated explicitly, but this is not expected to affect the estimates of risk appreciably. If necessary, this assumption can be re-examined if more detailed analysis is warranted.

3 Data Evaluation for Constituents of Interest

3.1 Constituent of Interest Evaluation

It is important that all site-related constituents are identified, and that the concentrations of are accurately quantified (EPA, 1994a). COIs to be carried forward to Tier II of the HHRA will be identified via the Tier I screening assessment.

Determination of whether a chemical constituent is present at the site depends upon, among other considerations, the selection of the constituents to be analyzed and the detection limits for the analytes. Determination of the spatial concentration distribution of detected and screened COIs is ideally determined via statistically designed grid-based sampling. In lieu of such sampling, the representativeness and precision/accuracy of samples must be determined. The representativeness and precision/accuracy of the samples collected in the RI for the different areas of the LIU will be re-examined to assess whether the data set is adequate for determining which COIs are present, and at what levels.

The COIs present in soil at the LIU have been deposited over time, primarily by deposition of airborne dust attributable to stockpiles. Thus, given the deposition mechanism for soil COIs, a gradient of COI concentrations in soil at the LIU may exist, with concentrations decreasing with increasing distance from the stockpiles. Statistical analysis will either support or not support this hypothesis.

3.2 Background Concentrations

The issue of ‘background’ is complex, but important. Any mine site has highly mineralized deposits in its natural state (i.e., pre-mining). Thus, environmental concentrations of metals may be high compared to other locations, even if there is no present or past mining activity. Comparisons of metal concentrations between mining-affected areas and non-affected areas (i.e., background or reference areas) are therefore difficult. The responsible party at a contaminated site (in this case, Chino) is not responsible for remediating areas of high background concentrations that have not been affected by the LIU stockpiles. At the LIU, upwind reference areas to the northwest and southwest of the Lampbright stockpile operations were chosen based on predominantly westerly wind directions (Arcadis 2011; Figure 3-4). However, it was found that areas to the north of the stockpiles were highly mineralized. For this reason, Neptune has chosen not to conduct *a priori* comparisons between site-related COI concentrations and LIU reference area concentrations for the purpose of screening mining-related COIs. Rather, the HHRA will estimate COI risks for the LIU site data, the LIU reference area data, and the reference area data from the STS IU (Gradient, 2008). The STS IU reference area was relatively non-mineralized in nature and may represent less-mineralized areas of the LIU.

3.3 Data Analysis

Neptune will assess the adequacy of site and reference area data based on completeness, comparability, and representativeness to make a final determination of whether the data are adequate to characterize the exposure areas at the LIU and are appropriate for use in the HHRA in terms of screening, estimating EPCs, and other purposes. Although the RI conducted a number of statistical analyses, additional analyses may be necessary for the purpose of the HHRA.

Data analysis activities will begin with exploratory data analysis, using such tools as box and whisker plots, scatter plots, bubble plots, etc. Graphical analyses are often the most important step in presenting and interpreting the data. These exploratory data analyses will be used to gain an understanding of the data and to investigate attributes related to the hypotheses of interest. These analyses will also help direct the ensuing statistical analyses by providing initial evidence of the likely results. The exact nature of statistical testing that might be conducted to confirm the findings of the exploratory data analyses will depend on the data and the distributional forms they support. Statistical testing could involve parametric as well as non-parametric techniques. Statistical analyses that are envisioned will cover spatial trends, correlation of key analytes, and testing of differences among parameters of interest.

3.4 Further Data Collection and Analysis

Task 2 of the Consulting and Professional Services agreement between Neptune and Chino indicates that Neptune should provide support to NMED for “the purpose of developing and implementing a supplemental sampling plan. . .to meet risk assessment needs”. Neptune has provided such advice to NMED in the form of comments on sampling and statistical methodology on the first and second draft RIs, and thus has satisfied this Task. Neptune also recommended further sampling to characterize background, as described in Arcadis (2011). If additional sampling and/or analysis is recommended based upon the results of the HHRA, such recommendations will be made in the context of the Uncertainty Assessment and Recommendations and Conclusions sections of the the HHRA (described below).

4 Exposure Assessment

4.1 Estimation and Use of Exposure Point Concentrations

EPCs may be calculated directly from the sampling data or from modeling results for different locations, times, and/or media based on the sampling data (e.g., contaminant concentrations in breathing zone dust or plants; see below). EPCs for soil and sediments will be calculated using all available, appropriate data for all COIs. Data qualified as 'estimated' will generally be used in the calculation of EPCs. For data qualified as 'not detected', appropriate statistical methods will be employed to best characterize COI concentrations.

The RI conducted a form of screening analysis, but the “human health decision criteria” employed in the RI (EPA Regional Screening Levels;

http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/) do not account for all exposure pathways of interest in the HHRA. The Tier I analysis employed in the HHRA will use maximum detected concentrations of all COIs in exposure models detailed below. Only those COIs that are risk-relevant will be carried forward into the Tier II analysis. Except where noted below, in the Tier II analysis both an average (mean) value and an upper-bound (95% UCL) value will be calculated to support assessment for 'average' and 'RME' conditions, respectively. If initial analyses indicate non-normal distribution of a COI, the data may be transformed prior to such calculation. Additional analysis of the statistical distribution of the data may be performed if appropriate to support calculation of average and upper-bound estimates.

Additional modeling is necessary to estimate EPCs in dust and in food. Examples of these models are presented in Appendix I.

4.2 Estimation of Contaminant Intake

4.2.1 Overview

Estimates of COI intake are typically expressed in terms of the daily amount of the COI entering a receptor's body (via ingestion or inhalation) divided by the weight of the individual. Contaminant intake is averaged over the period of time for which exposure occurs, although this period is often split into 'childhood' and 'adulthood'. The basic approach to estimating intake of contaminants is to use generic intake rates (soil ingestion, inhalation, etc.) and estimate the time on-site during which an individual takes in contaminants at these rates. The HHRA will employ RME estimate assumptions for generic intake rates. If necessary to support risk management decisions, activity-specific intake rates and behavioral patterns may be specified later for critical pathways.

4.2.2 Exposure Estimation

The exposure equations used in the HHRA are based upon the general equations presented in EPA's *Risk Assessment Guidance for Superfund* (EPA 1989). These equations are used to estimate the amount of contaminant entering the body via specific exposure routes.

Note that exposure to soil or sediment via ingestion, dermal contact, and/or dust is dependent upon how much yearly exposure occurs in the area of interest; thus equations may be modified to reflect this. The same applies to the proportion of the year that soil or sediment may be frozen or snow covered. Bioavailability (i.e., ability of a compound to be absorbed) adjustments may also be performed.

Examples of exposure equations include:

Ingestion of Soil/Sediment

$$\text{Intake} = \frac{C \times CR \times EF \times ED}{BW \times AT}$$

where,

Intake = chronic daily constituent intake (mg/kg body weight/d)

C	=	constituent concentration in exposure medium (mg/kg soil)
CR	=	contact rate (mg soil/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	time over which exposure is averaged for experiencing adverse effect (d)

Surface Water Ingestion

$$\text{Intake} = \frac{C \times CR \times EF \times ED}{BW \times AT}$$

where,

Intake	=	chronic daily constituent intake (mg/kg body weight/d)
C	=	constituent concentration in exposure medium (mg/L water)
CR	=	contact rate (L/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	time over which exposure is averaged for experiencing adverse effect (d)

Dermal Contact with Soil/Sediment

$$\text{Intake} = \frac{C_s \times ABS \times SA \times AF \times EF_{\text{derm}} \times ED \times CF}{BW \times AT}$$

where,

Intake	=	chronic daily constituent intake (mg/kg body weight/d)
C _s	=	constituent concentration in soil (mg/kg soil)
ABS	=	dermal absorption fraction (dimensionless)
SA	=	exposed skin surface area (cm ²)
AF	=	soil adherence factor (mg/event)
EF _{derm}	=	exposure frequency for dermal contact with soil (event/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
CF	=	conversion factor (mg/kg)
AT	=	averaging time (d)

Ingestion of Fruits or Vegetables

$$\text{Intake} = \frac{C_s \times K_{p-s} \times IR_p \times F_p \times EF \times ED \times CF}{AT}$$

where,

Intake	=	chronic daily constituent intake (mg/kg body weight/d)
C_s	=	constituent concentration in soil (mg/kg soil)
K_{p-s}	=	plant – soil concentration ratio (mg/kg plant per mg/kg soil)
IR_p	=	plant ingestion rate (g/kg body weight/d)
F_p	=	fraction of plants ingested that are grown in affected area
EF	=	exposure duration (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (kg/g)
AT	=	averaging time (d)

Ingestion of Meat or Eggs

$$\text{Intake} = \frac{C_s \times TF_{s-m} \times ((UR_f \times K_{f-s}) + UR_s) \times IR_m \times F_m \times EF \times ED \times CF}{AT}$$

where,

Intake	=	chronic daily constituent intake (mg/kg body weight/d)
C_s	=	constituent concentration in soil (mg/kg soil)
TF_{s-m}	=	soil-to-meat transfer factor (mg/kg meat per mg/d)
UR_f	=	uptake rate of feed by animal (kg/d)
K_{f-s}	=	feed – soil concentration ratio (mg/kg feed per mg/kg soil)
UR_s	=	uptake rate of soil by animal (kg/d)
IR_m	=	meat ingestion rate (g/kg body weight/d)
F_m	=	fraction of meat ingested that is raised in affected area
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
CF	=	conversion factor (kg/g)
AT	=	averaging time (d)

Although the subscript in this equation refers to meat, this equation also applies to intake of a constituent from eggs. The contact rate of an animal with soil may also be fractionated to apportion less than 100% of either soil or feed to the area for which the constituent concentration term applies. For cattle, constituent uptake will be assumed to potentially occur via both grazing plants (feed) and directly ingesting soil in an affected area. For chickens, constituent uptake will be assumed to potentially occur only *via* directly ingesting soil in an affected area; i.e. it is assumed that store-bought feed is used.

Inhalation of Dust

$$\text{Intake} = \frac{C_s \times \text{InhR} \times \text{ET} \times \text{EF} \times \text{ED}}{\text{PEF} \times \text{BW} \times \text{AT}}$$

where,

Intake	=	chronic daily constituent intake (mg/kg body weight/d)
C_s	=	constituent concentration in soil (mg/kg soil)
InhR	=	inhalation rate (m^3 /hour)
ET	=	exposure time (hr/d)
EF	=	exposure frequency (d/yr)
ED	=	exposure duration (yr)
PEF	=	particulate emission factor (calculated; m^3 /kg soil)
BW	=	body weight (kg)
AT	=	averaging time (d)

The PEF is in effect the volume of air occupied by one kilogram of suspended soil or dust, and is the output of a dust resuspension model discussed previously (also see Appendix I). If air samples are collected to directly measure concentrations in suspended dust, the PEF term is eliminated and the COI concentration term may be expressed as mg/m^3 of air.

These equations may be modified to reflect the final exposure pathways evaluated in the HHRA.

4.2.3 Exposure Variables

RME upper-bound values that are consistent with EPA's *Standard Default Exposure Factors* (EPA, 1991a) and the *Exposure Factors Handbook* (EPA, 2011a) will be employed in exposure equations. These may be changed in the HHRA if additional or improved information become available. Some of the variable values may change depending upon whether children (vs. adults) are evaluated in particular scenarios.

As previously indicated, a probabilistic analysis will use distributions for the values of exposure variables rather than a single point estimate. If this type of analysis is conducted, distributions will be defined and documented.

Exposure variable values are not presented at this time, as appropriate values will depend upon the exact nature of final exposure pathways and equations evaluated in the HHRA.

5 Toxicity Assessment

Regulatory constituent-specific toxicity values for evaluating cancer and non-cancer endpoints are referred to as slope factors (SF) and reference doses (RfD), respectively. These are essentially 'conversion factors' applied to intake estimates. The SF (in $[mg/kg-d]^{-1}$) is derived using the assumption that there is no threshold of exposure below which a carcinogenic (cancer-causing) response may not occur; i.e., there is no 'safe' exposure level to a carcinogen. This is a conservative assumption. The probability of cancer induction is typically modeled as linearly related to the degree of exposure (i.e., a direct relationship), which again is a conservative assumption.

The RfD has been developed for non-cancer causing toxic agents based on the concept that a threshold dose exists below which adverse effects are not likely to be observed. Unlike the SF, thresholds of effect are acknowledged, and the probability of experiencing an adverse effect is not directly related to the intensity of exposure in a continuous

manner. RfDs often have multiple ‘safety factors’ applied (especially those based upon animal studies), and thus represent conservative estimates of ‘safe’ levels.

The primary source of toxicity values used in the HHRA will be EPA’s Integrated Risk Information System database (IRIS; EPA, 2011b). Only toxicity criteria published in IRIS have gone through peer-review and EPA-consensus-review processes. The second tier of toxicity criteria are the provisional peer-reviewed toxicity values (PPRTV) published by the National Center for Environmental Assessment (NCEA) in EPA’s Office of Research and Development. These values are developed on a chemical-specific basis when requested by EPA’s Superfund program, but the documentation for them is generally not citable. The third tier of references include values published in EPA’s *Health Effects Assessment Summary Tables* (HEAST; EPA, 1997) and other sources such as California EPA and the Agency for Toxic Substances and Disease Registry.

With the exception of an inhalation toxicity value for copper, proposed toxicity values will be obtained from the EPA sources described above. A copper inhalation toxicity value was derived by Gradient Corporation for the Hurley Soils IU Human Health Risk Assessment (Gradient, 1998) and will be used in the LIU HHRA.

Toxicity values are specified separately for the ingestion and inhalation exposure routes. The relative bioavailability of a compound taken into the body via ingestion or inhalation may be evaluated. In general, bioavailability of a compound in soil is expected to be lower than from food or water due to the time required for the compound to desorb (become ‘unattached’) from a soil particle and/or diffuse from within pores in the soil particle. Some fraction of a compound adsorbed onto soil may be permanently adsorbed (‘attached’), or else desorb at so slow a rate as to be effectively irremovable. The degree of bioavailability from soil is affected by factors such as chemical form, particle diameter, geochemical factors, and the nutritional status of an individual. Where available, scientifically documented bioavailability fractions will be applied in the HHRA.

Although EPA publishes oral toxicity values directly as SFs and RfDs, inhalation toxicity values are published as unit risks (URs) and reference concentrations (RfCs) (in units of $(\text{mg}/\text{m}^3)^{-1}$ and $(\mu\text{g}/\text{m}^3)^{-1}$, respectively). Inhalation RfD and SF values have traditionally been calculated from the published UR and RfC values assuming (as applicable) average inhalation rate, water ingestion rate, and body weight values of 20 m^3 / day, 2 L / day and 70 kg, respectively. In some cases it is necessary to modify these assumptions to allow for different circumstances.

Lead is addressed differently than other metals by EPA, as it exhibits complex dynamics in the human body. EPA has recommended a residential screening level for lead in soil of 400 mg/kg, derived using a biokinetic model. The 400 mg/kg screening level was developed such that a typical child would have no more than a 5% chance of having a blood lead level exceeding 10 $\mu\text{g}/\text{dl}$, a level thought to be associated with health effects in children (EPA, 1994b). Site-related residential exposures contributing to the 400 mg/kg screening level include soil ingestion from the yard and indoor ingestion of house dust contaminated with soil. In addition to these site-related exposures, the 400 mg/kg screening level incorporates background levels of lead exposure from non-site related sources including ambient air, drinking water, and diet. These background exposures

were defined using “national averages, where suitable, or typical values” (EPA, 1994b). For the purpose of the HHRA, the 400 mg/day value has been adopted for assessing potential risks to children in both the residential scenario and the child recreational scenario. No soil or sediment values in the LIU exceed this criterion; therefore lead will not be addressed explicitly in the HHRA.

Toxicity values to be employed in the HHRA are presented in Appendix II. These values will be reviewed (e.g., for updates) prior to risk calculations.

6 Risk Characterization

6.1 Overview

In risk characterization, site-related COI exposures and toxicity values are combined to produce estimates of incremental lifetime cancer risk (ILCR) and non-cancer hazard. These estimates are then compared with ‘acceptable’ levels, as determined by regulatory guidance, precedent, and discussion among involved parties (see below).

‘Background’ risk and hazard associated with COIs at reference areas previously discussed will also be estimated for comparative purposes.

6.2 Estimation of Incremental Lifetime Cancer Risks

ILCR is a concept used by EPA in environmental HHRA (EPA, 1989; 1990).

‘Incremental’ is defined as the risk associated with a specific exposure that is increased over all-cause cancer risk, which is approximately 1 in 3 (or 33%) over an average lifetime. SFs specific for COIs and exposure routes are used to convert estimated daily intake over an exposure period to ILCR, as:

$$\text{ILCR} = \text{Intake} \times \text{SF}$$

where,

ILCR	=	lifetime incremental cancer risk (dimensionless)
Intake	=	chronic daily intake (mg/kg-d)
SF	=	slope factor (mg/kg-d) ⁻¹

ILCR estimates will be calculated for individual COIs. Typically these are summed across both route of intake and exposure pathways for a given scenario. However, the potential additivity of pathways, and particularly whether a RME could occur for two or more pathways simultaneously for an individual, will be evaluated before pathway risks are summed.

ILCRs across individual COIs will also be summed to estimate a total ILCR, in accordance with guidance presented in EPA (1989). However, there are a number of issues associated with this. ‘Cancer’ is not one disease, but hundreds; each having a unique clinical profile and natural history (e.g., liver cancer is a very different disease than skin cancer). Additionally, there are differences in the derivation and level of confidence associated with individual SFs. These and similar issues will be addressed in the uncertainty assessment section of the HHRA.

The final ILCR that may be acceptable will be determined by the involved parties. The ILCR *de minimus* (i.e., minimal or not measurable in a public health study) range of 1×10^{-4} to 1×10^{-6} (0.0001 to 0.000001) described in the National Contingency Plan (EPA, 1990) has been used by involved parties as a decision aid. NMED has defined 1×10^{-5} (0.00001) as a target for development of its Soil Screening Levels (SSLs) (NMED, 2009). For context, the total lifetime risk of cancer to a receptor exposed to a COI that is associated with a 1×10^{-4} ILCR would be approximately 0.33 plus 0.0001, or 0.3301.

Only risk-relevant COIs screened in the Tier I assessment will be carried forward to the Tier II assessment. In Tier II, Neptune proposes that COIs with ILCRs smaller than 1×10^{-4} will not be subject to further analysis or investigation (consistent with EPA, 1991b), whereas COIs with ILCRs greater than this level may be subject to further analysis and investigation. The information provided in the uncertainty assessment regarding the confidence and potential biases associated with the risk estimates should be used to inform ultimate decisions.

6.3 Estimation of Non-Cancer Hazards

RfDs specific for COIs and intake routes are used to convert estimated daily intake over an exposure period to a HQ. Unlike an ILCR, a HQ does not reflect the probability of an effect occurring. However, larger values of HQ can be associated with potentially increased severity of effects. The equation for calculating the HQ is:

$$HQ = \frac{\text{Intake}}{\text{RfD}}$$

where,

HQ	=	hazard quotient
Intake	=	chronic daily intake (mg/kg-d)
RfD	=	reference dose (mg/kg-d)

The RfD is assumed to be linearly related to HQ in this equation. HQs above 1.0 (i.e., the estimate intake level exceeds the RfD) are of potential concern (consistent with EPA and NMED guidance). The potential for additive non-cancer effects across two or more COIs will be evaluated in the HHRA only in cases where the toxic effects of the COIs are similar. The sum of two or more HQ values is referred to as a Hazard Index (HI). A HI value exceeding 1.0 may be of concern even if the HQs for all individual COIs is below 1.0. If RfD values corresponding to different exposure duration are used in the HHRA, HI values will be tabulated separately on this basis as well. Depending on the magnitude of the HI value and the number of risk-driving COIs contributing to the HI, HI values may also be tabulated separately based on target organs associated with the toxic effect for the RfD. Regardless, potential additivity for non-cancer effects will be addressed in the uncertainty assessment section of the HHRA.

HQs will be summed across both route of intake and exposure pathways for a given scenario. The potential additivity of pathways, and particularly whether a RME exposure

could occur for two or more pathways simultaneously for an individual, will be evaluated before pathway hazards are simply summed.

The HQ or HI value that is generally indicative of the potential for adverse health effects is 1.0. The information provided in the uncertainty assessment regarding the confidence and potential biases associated with HQ or HI estimates will be used to inform the involved parties in determining an appropriate decision if HQ or HI values are above 1.0.

7 Uncertainty Assessment

The goal of the uncertainty assessment is to provide the involved parties with useful information regarding the level of confidence in the risk estimates and the direction and magnitude of potential biases. Uncertainty assessment can be qualitative and/or quantitative in nature. In general, the uncertainty assessment will include model uncertainty and variable uncertainty.

Model uncertainty addresses the influence of the model form and boundary conditions on the risk estimates. Variable uncertainty addresses the influence of variable values on the risk estimates. Within the variable uncertainty category a further distinction is made between uncertainty associated with spatial variability and transport of COIs, and uncertainty in the behavior and characteristics of the receptor population. It will be assumed that there is no correlation between COI distribution and receptor characteristics.

There are several aspects for which model uncertainty may be important in interpreting the risk calculations. For the purpose of this HHRA, these sources of uncertainty will be discussed qualitatively. For example, the supporting transport models (plant uptake, animal uptake, dust resuspension) proposed for predicting EPCs in unsampled media are generally consistent with what are often called 'screening-level' calculations. There may be known conservative biases associated with these models due to simplifying assumptions that minimize data needs. Similarly, conservative assumptions are typically incorporated into the models used by EPA to generate toxicity values, although these assumptions may not be 'biases' if the subpopulation of interest has high sensitivity to one or more toxicants. Finally, the types of activities that may occur at various locations in the LIU and the potential scale of these activities may be important aspects of model uncertainty.

Variable uncertainty is often addressed quantitatively. One method is to compare RME estimates with 'average' estimates, but as previously discussed this is not as informative as more formal methods. As there is a high degree of confidence that risks under Tier II will be overestimated (as opposed to underestimated), there may be no need for a quantitative analysis if risks fall within acceptable levels. However, if risks exceed acceptable levels, then a simple probabilistic analysis is warranted only for those COIs, receptors, and exposure pathways of concern. A probabilistic analysis will involve defining statistical distributions for exposure variables, and combining these distributions in exposure equations using Monte Carlo simulation, a means of randomly drawing from each distribution many times so that distributions of exposure and risk result (Cullen and Frey, 1999). This will allow determination of average estimates of risk and where RME estimates fall on the distributions, as well as determination of which variables contribute

most to overall uncertainty via ‘sensitivity analysis’. Sensitivity analysis informs additional information collection (if required) and provides focus upon the variables that are ‘driving’ the risk results.

Note that such probabilistic analyses will still be conservatively biased to some extent, as they will incorporate deterministic SFs and RfDs, which in turn incorporate conservative assumptions and safety factors. Determination of uncertainty associated with SFs and RfDs will likely be beyond the scope of this HHRA, although this uncertainty may be addressed in development of RAC, if required.

8 Recommendations and Conclusions

A section on recommendations and conclusions will be provided in the HHRA to summarize the results of the risk characterization and interpret these results relative to various decision alternatives. Further information collection recommendations, if any, will be described. This section will be developed in collaboration with the involved parties to facilitate integration of this assessment with other environmental investigations associated with the AOC and other regulatory vehicles. Recommendations for development of RAC, if required, will also be made.

9 Reporting

Brief technical memoranda will be provided for review by NMED, EPA, and Chino for the following components of the HHRA:

- Conceptual site model (CSM);
- Data evaluation for COIs;
- Exposure assessment;
- Toxicity assessment;
- Risk characterization;
- Uncertainty assessment; and,
- Recommendations and conclusions.

Upon review and finalization, these memoranda will provide the basis for an internal draft HHRA report (to be provided to NMED, EPA, and Chino), and an external draft HHRA report (to be provided to all involved parties). Both draft HHRA reports will be written in lay language where possible, and all technical terms and approaches will be fully explained.

10 Schedule

A detailed schedule is not possible at this time, as completion and finalization of the different components of the HHRA are contingent upon the availability and review schedule of NMED and Chino. However, given prompt review and resolution the following schedule is anticipated; given a start date of January 3, 2012 (subject to change):

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- Delivery of the internal draft HHRA is anticipated April 30, 2012.
 - Delivery of the external draft HHRA is anticipated 3 weeks after receipt and resolution of comments on the internal draft.
 - Delivery of the final HHRA is anticipated 3 weeks after receipt and resolution of comments on the external draft.

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Appendix I: Dust and Biota Exposure Point Concentration Models

Dust

As ambient concentrations of PM10 (respirable particulates having a diameter of approximately 10 µm and less) are not routinely monitored at the LIU, these concentrations must be modeled. EPCs in air above contaminated soils in the LIU will initially be calculated using a screening-level soil resuspension and air dispersion model. The specific model that will be used for these screening calculations is EPA's particulate emission factor (PEF) model. The PEF model for wind erosion can be used to estimate annual average concentrations PM10 in ambient air (EPA, 1996; EPA, 2002). The PEF model has two components. The first component is a particulate emission model for wind erosion of soils. The second component is an atmospheric dispersion term (Q/C_{wind}) that relates air concentrations to particulate emissions.

The ratio of the concentration of respirable particulates in air to the particle flux from the ground is represented in the PEF model by the Q/C_{wind} term, which is defined as the inverse of this ratio. This term is derived from atmospheric dispersion modeling using the Industrial Source Complex air dispersion model in short-term mode (ISCST3) for a variety of source sizes and meteorological conditions.

Q/C_{wind} calculations based on a least-squares curve fit of site size and dust concentration were performed by EPA for 29 sites and documented in Appendix D of EPA (2002). The resulting equation, provided as Exhibit D-1 of EPA (2002) is:

$$Q/C_{wind} = A \times \exp \left[\frac{(\ln A_{site} - B)^2}{C} \right]$$

where

Q/C_{wind} = inverse of the mean particulate concentration at the center of a square source; in area per unit particulate flux (g/m²-sec per kg/m),

A_{site} = area of site (acres),

A, B, C = curve fitting constants; and,

exp = exponent applied to the base of the natural logarithm e .

Values of the constants A, B, and C for a local meteorological station (e.g., Albuquerque, NM) will be used to represent conditions in the LIU (EPA 2002; Exhibit D-2). The area(s) pertaining to soil EPCs will be defined in the HHRA.

The wind erosion component of the PEF model is comprised of the remaining terms in the PEF equation, which is defined in EPA (1996; 2002) as:

$$PEF_{wind} = Q / C_{wind} \times \frac{3600 \text{ sec/hr}}{0.036 \times (1 - v) \times (U_m / U_{t-7})^3 \times F(x)}$$

where:

PEF_{wind} = particulate emission factor for wind-generated erosion (m^3/kg),

Q/C_{wind} = inverse of the mean particulate concentration at the center of a square source area per unit particulate flux (g/m^2 -sec per kg/m^3),

U_m = mean annual windspeed (m/sec),

U_{t-7} = equivalent threshold value of windspeed at 7 m height (m/sec),

v = fraction of vegetative cover (dimensionless); and,

$F(x)$ = function dependent on U_m / U_{t-7} (dimensionless).

A mean annual wind speed from the nearest meteorological station will be used in the HHRA. The fraction of vegetative cover will be estimated from information obtained for the ecological risk assessment for the LIU. EPA (1996; 2002) default values will be used for the remaining PEF equation variables in the screening calculations.

Depending upon the results of pathway and uncertainty analyses, and upon the calculated risk values, site-specific dispersion modeling may subsequently be employed in the Tier II assessment for refining the estimate breathing zone particulate concentrations in the LIU.

Garden produce, livestock, and game

EPCs in home-raised produce will be calculated based on soil EPCs using published regression models or produce-soil concentration ratios (K_{p-s}) for plants. Regression models and values of K_{p-s} will be obtained from Bechtel-Jacobs (1998). These models will be employed to estimate produce concentrations related to root uptake of metals in plants growing on contaminated soil. If a regression model is unavailable, median (i.e., 50th percentile) linear K_{p-s} values published in Bechtel-Jacobs (1998) will be used.

In addition to modeled estimates of metals concentrations in plant tissues, analytical data from the site-wide ecological risk assessment may also be used to support estimates of plant tissue concentrations. Specifically, paired samples of plant tissue and soil may be used if available. Actual metal concentrations across the active root zone may only be approximated with such soil data, and the native vegetation sampled may not accumulate metals in a manner similar to produce or forage plants. However, as a line of evidence for qualifying modeled plant concentrations these data may still be useful.

Models for estimating EPCs in beef and game are published in EPA's Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (EPA, 2005). For ranched cattle and game, the calculation of fresh-weight EPCs in muscle tissue accounts for the sum of contaminant intake by the animal via direct soil ingestion while grazing and indirect intake via wild grasses and other plants growing on contaminated soil. The calculation is:

$$EPC_{meat} = C_{soil} \times TF \times [(UR_g \times K_{f-s}) + (UR_s \times B_s)] \times MF$$

where:

EPC_{meat} = concentration of metal in beef or game meat (mg/kg)

C_{soil}	= concentration of contaminant in soil (mg/kg)
TF	= feed-beef transfer factor for cattle or game (mg/kg fresh meat per mg/day)
UR_g	= uptake rate of forage plants by cattle or game, dry weight (kg/day)
K_{f-s}	= forage-soil concentration ratio (mg/kg dry grass per mg/kg soil)
UR_s	= uptake rate of soil by cattle or game (kg soil/d)
B_s	= soil bioavailability factor (dimensionless)
MF	= metabolism factor (dimensionless).

Values for model exposure variables will be obtained from EPA (2005), Wang et al. (1993), Baes et al (1984), and other references as appropriate. EPCs for metals in chicken tissue and eggs are calculated based on transfer factors from EPA publications or other sources in a manner similar to that described for cattle and game. EPA (2005) addresses transfer factors for chicken tissue and eggs for organic chemicals, but for only a limited number of metals, including mercury compounds, cadmium, selenium, and zinc. Other references for poultry transfer factors include Ng et al. (1982) and IAEA (2010). Although poultry transfer factors relate to uptake from feed, they will be applied to uptake of metals from soil in the risk assessment. It is assumed that chicken feed is store-bought, rather than produced from grain grown onsite, and that exposure to soil contaminants for free-range chickens is a result solely of their foraging habits.

Appendix II: Toxicity Values

Constituent	Oral RfD		Inhalation RfC		Oral SF		Inhalation Unit Risk	
	(mg/kg-d)	Source	(mg/m ³)	Source	(mg/kg-d) ⁻¹	Source	(μg/m ³) ⁻¹	Source
Aluminum	1.0E+00	PPRTV	5.0E-03	PPRTV				
Antimony ¹	4.0E-04	IRIS	2.0E-04	IRIS				
Arsenic	3.0E-04	IRIS	1.5E-05	CalEPA	1.5E+00	IRIS	4.3E-03	IRIS
Barium	2.0E-01	IRIS	5.0E-04	HEAST				
Beryllium	2.0E-03	IRIS	2.0E-05	IRIS			2.4E-03	IRIS
Boron	2.0E-01	IRIS	2.0E-02	HEAST				
Cadmium	1.0E-03 (diet) 5.0E-04 (H ₂ O)	IRIS	2.0E-05	CalEPA			1.8E-03	IRIS
Chromium (III) ²	1.5E+00	IRIS						
Chromium (VI) ²	3.0E-03	IRIS	1.0E-04	IRIS	5.0E-01	NJDEP	8.4E-02	IRIS (adj)
Cobalt	3.0E-04	PPRTV	6.0E-06	PPRTV			9.0E-03	PPRTV
Copper	4.0E-02	HEAST	2.4E-02	Gradient 1998				
Iron	7.0E-01	PPRTV						
Manganese ³	1.4E-01 (diet) 2.4E-02 (other)	IRIS	5.0E-05	IRIS				
Mercury ⁴	3.0E-04	IRIS	3.0E-05	CalEPA				
Molybdenum	5.0E-03	IRIS						
Nickel ⁵	2.0E-02	IRIS	9.0E-05	ATSDR			2.6E-04	CalEPA
Selenium	5.0E-03	IRIS	2.0E-02	CalEPA				
Silver	5.0E-03	IRIS						
Thallium ⁶	1.0E-05	PPRTV						
Vanadium ⁷	5.0E-03	IRIS						
Zinc	3.0E-01	IRIS						

General Notes:

Most toxicity values are consistent with the November 2011 EPA Regional Screening Level (RSL) summary tables (http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm).

RfD= reference dose; RfC= reference concentration; SF= slope factor; PPRTV= provisional peer-reviewed toxicity value; IRIS= Integrated Risk Information system; CalEPA= California Environmental Protection Agency; HEAST= Health Effects Assessment Summary Tables; NJDEP= New Jersey Department of Environmental Protection; ATSDR= Agency for Toxic Substances and Disease Registry.

Specific Notes:

1. Oral RfDs published for metallic form (4E-04 mg/kg-d), tetroxide (4E-04 mg/kg-d), and pentoxide (5E-04 mg/kg-d). Values are similar; 4E-04 mg/kg-d selected. Inhalation RfC (2E-04 mg/m³) published for antimony trioxide. Trioxide form created in the atmosphere by reaction with atmospheric oxidants and oxidation also occurs in aerobic surface soils (http://ntp.niehs.nih.gov/ntp/htdocs/Chem_Background/ExSumPdf/Antimonytrioxide.pdf).
2. Inhalation unit risk value of 1.2E-02 per µg/m³ in IRIS assumes the ratio of Cr (III) to Cr (VI) in air is 6:1. The value of 8.4E-02 per µg/m³ is adjusted by a factor of 7 to pertain to only Cr (VI) (Section 5.6 of RSL User's Guide). Chromium (VI) is listed as having a mutagenic mode of action when administered by drinking water in EPA's RSL table and therefore requires use of age-dependent adjustment factors for early-life exposures (http://www.epa.gov/spc/pdfs/CGIWGCommunication_II.pdf).
3. Non-dietary oral RfD includes adjustment for subtraction of dietary manganese contribution to daily intake and modifying factor of 3 for non-food sources (Section 5.6 of RSL User's Guide).
4. As mercuric chloride and other mercury salts.
5. As nickel soluble salts. Nickel oxides and sulfides are associated with combustion, smelting, and refining sources (<http://www.atsdr.cdc.gov/toxprofiles/tp15-c6.pdf>).
6. Value for thallium soluble salts is based on information in an appendix to the PPRTV manuscript.
7. Value based on vanadium pentoxide (V₂O₅) oral RfD of 9E-03 mg/kg-d, adjusted for the contribution of the mass of the oxide ion (9E-03 × 56% = 5.04E-03) (Section 5.6 of RSL User's Guide).